

*Suffolk County Vector Control & Wetlands
Management Long Term Plan & Environmental
Impact Statement*



**Task 5: Data Collection
Groundwater Discharge and
Long Island Salt Marshes**

Prepared for:

**Suffolk County Department of Public Works
Suffolk County Department of Health Services
Suffolk County, New York**

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June 2005

**SUFFOLK COUNTY VECTOR CONTROL AND WETLANDS MANAGEMENT
LONG - TERM PLAN AND ENVIRONMENTAL IMPACT STATEMENT**

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Acronyms and Abbreviations

GPS	global positioning system
kg/l	kilograms per liter
km	kilometer
m	meter
MDS	multi-dimensional scaling
MGD	millions of gallons per day
MUS	marshland upwelling system
NOAA	National Oceanic and Atmospheric Administration
NWR	National Wildlife Refuge
PCA	principal component analysis
ppt	parts per thousand
SCDHS	Suffolk County Department of Health Services
USGS	US Geological Survey

EXECUTIVE SUMMARY

Salt marshes exist where the shore meets the estuary. This is also the area where Long Island's groundwater aquifers discharge. Due to mixing processes – between fresh water from rainfall, run-off, and/or groundwater, and salt water from the estuary – salt marshes exist in brackish environments. Salt marshes are also characterized by the presence of a salt water table just beneath the surface of the marsh. This salt water “aquifer” is generally understood to be perched above the more general groundwater aquifer below, which is usually thought to be fresh.

The dynamics between tides, rainfall, any run-off inputs, and potential discharges of fresh groundwater are clearly important to many processes that occur in a marsh. Inputs of nutrients and the overall salinity of the marsh environs are thought to be key to modern-day changes occurring in marshes, such as the invasion of *Phragmites australis*, the balance between high marsh and low marsh plants, and potentially the overall maintenance of marsh health. Therefore, it is important to understand the relationship between groundwater discharge and salt marshes.

Groundwater discharge to estuaries has been the subject of increasing research efforts over the past twenty years. Long Island has been a hotbed of these efforts. Suffolk County Department of Health Services has, in fact, been a nation-wide leader in the development of new technologies to measure submarine discharges, and to interpret the data produced by these experiments. One aspect of this study is to determine if it would be beneficial to apply these kinds of new experimental devices in the salt marsh to assist in determining processes there.

There are many different theories regarding the relationship between the salinity of the salt water salt marsh aquifer, groundwater, and other factors. Several are concerned with the way groundwater may percolate through the marsh peat, and the forces that control this process; others believe groundwater inputs to this system are negligible. Based on several years of data collected at the Wertheim National Wildlife Refuge Open Marsh Water Management Demonstration Project, Cashin Associates, PC attempted to determine which theory seems to fit best for the South Shore mainland marshes, which are likely to be a focus of water management efforts by Suffolk County Vector Control. Although the large data set generated there illuminated the conditions at the marsh at various times, the data did not generally support any

theory in particular. Rather, these data suggested that the salinity of the salt marsh water table is the product of complex and interlinked processes.

1. Introduction

1.1 General Concepts Relating to Groundwater Discharge

Groundwater discharges from an unconfined aquifer in locations where the head elevation for the aquifer is greater than its corresponding sediment surface elevation (Freeze and Cherry, 1979). This creates a continuum between groundwater and surface waters (USEPA, 2000). On Long Island, where the Upper Glacial aquifer is intersected by the ground surface, streams and ponds occur.

US Geological Survey (USGS) modeling suggests 32 percent of annual recharge discharges to stream systems in Suffolk County (Table 1) (Buxton and Smolensky, 1999). Inconsistencies in the data in Table 1 reflect transfers between the geographical areas, and rounding. This discharge to streams impacts the water table, causing lower head pressures in the near vicinity of streams. The measurable effect, especially for smaller streams, is often extremely local. At Connetquot Brook, the difference in heads was detectable only 30 vertical feet below the creek and approximately the same distance from each bank (Prince et al., 1989). However, other modeling has suggested that even modest streams can drain large portions of individual watersheds (the fresh water portion of Meetinghouse Creek collected 25 percent of the recharge of that area) (Schubert, 1999).

Table 1. Groundwater budget for 1968-1983 conditions on Long Island (millions of gallons per day – [MGD])

	Recharge	Discharge			
County	(Precipitation & returned water)	(Pumpage)	(Stream)	(Shore)	(Subsea)
Kings & Queens	136	77 (57%)	12 (9%)	56 (41%)	2 (1%)
Nassau	346	185 (53%)	55 (16%)	82 (24%)	14 (4%)
West Suffolk	339	87 (26%)	123 (36%)	126 (37%)	25 (7%)
East Suffolk	472	58 (12%)	135 (29%)	239 (51%)	17 (4%)
All Suffolk	811	145 (18%)	258 (32%)	365 (45%)	42 (5%)
Total	1,293	407 (31%)	325 (25%)	503 (39%)	58 (4%)

(adapted from Buxton and Smolensky, 1999)

Research on coastal plain streams fed by groundwater indicates that the upper stretches are often receiving recently recharged groundwater (which thus is from the immediate vicinity of the stream). This changes for downstream reaches, where discharge from the banks or stream bottom close to the banks may have been recharged locally, but discharges into the central portions of the stream bottom often have long aquifer residence time, and thus may be from areas of the watershed that are not particularly close to the banks of the stream (Modica et al., 1998). It should also be noted that discharge in a stream is spatially variable, as certain parts of the stream may have prolific sources of water at near point sources (“springs”), greater than average sources of water that may be compact in area or also more widespread (“preferential flowpaths”), or areas of low discharge, no discharge, or even recharge. Underlying geological conditions, sometimes at some depth below the stream, determines the discharge conditions (Conant, 2004). In addition, although not as large an issue on Long Island where streams generally are less sinuous than sometimes found elsewhere, changes in flow direction and meanders can influence where and how much groundwater discharges through a streambed (Winter, 2000).

Discharge through the streambed is thought to impact water quality, although many of these processes are not well understood (Conant, 2000). Conant classified the classes of potential reactions as destructive, where the compound is irreversibly changed, and non-destructive, where the concentration of the compound is altered by what are generally reversible processes. The zone where these reactions occur is generally defined as the hyporheic zone (Winter, 2000).

1.2 Submarine Groundwater Discharge

At the shoreline, the elevation of the aquifer is greater than the surface of the sediment. Groundwater discharges through the salt water interface. USGS modeling suggests that 45 percent of recharge discharges at or near the shoreline in Suffolk County (Buxton and Smolensky, 1999). This phenomenon has begun to receive attention.

Salt water is denser than fresh water. The average salinity of the oceans is said to be about 35 parts per thousand (ppt). This makes its density between 1.022 and 1.028 kilograms per liter (kg/l). (Reilly and Goodman, 1985).

In advection-dominated systems, the interface between fresh water and salt water is generally considered to be sharp. In density-flow or dispersion dominated systems, then a large area of

mixing is usually anticipated. These very simplified depictions do not take into account fine-scale phenomenon, nor do they account for aquifer heterogeneities. As a matter of fact, all systems have something of a zone of mixing. A circulation pattern develops where salt water mixes with fresh water, and moves seaward from the interface. This causes saltier water to move towards the interface (Reilly and Goodman, 1985).

The Ghyben-Herzberg formula describes the relationship between the depth of the salt water interface, and the head of the fresh water aquifer above sea level. It depends on the density difference between the fluids (Freeze and Cherry, 1979). The equation has been modified to account for anisotropism (which results in a flattening of the fresh water lens) (Nemickas and Koszalka, 1982). The equation also assumes that the fresh water head at the shoreline is zero, whereas the discharge of fresh water into the salt water aquifer shows that is not accurate (Reilly and Goodman, 1985). Glover (1959) accounted for this process better, allowing for discharge at the shore and balancing pressures in the system, albeit in a much more mathematically complex way than the elegant and simple Ghyben-Herzberg solution.

Bokuniewicz (1980) was one of the first to quantify aquifer discharge to the nearshore environment. His studies suggested that most of this submarine discharge from the Upper Glacial aquifer occurred within a hundred feet or so of the shore. Bokuniewicz (1992) developed an analytical solution of discharge, where it was a function of vertical and horizontal hydraulic conductivities, aquifer thickness, and distance to the shoreline, recognizing that factors such as salt fingering and secondary convection due to density differences would make direct measurements of the fluxes different from modeled flows.

Follow-up work has shown that the discharge rates are highly variable. They are a function of tidal cycles and sediment characteristics (Robinson et al., 1998). Small variations in hydraulic conductivity may be very important, and in addition, small lenses of more impermeable material (such as meter-thick layers of peat in irregular depositions) can have very large impacts on the distribution of underlying fresh and salt water reservoirs (Krantz et al., 2004). Some work has found that there are important time lags between maximal discharges and lowest tides (Rapaglia and Bokuniewicz, 2004), probably as a result of the propagation of tidal influences slowly being overcome by the aquifer hydraulic gradient (Paulsen et al., 2004). There is a spatial element to tidal influences, as the location of greatest discharge has also been observed to move seaward

with ebbing tides (Urish and McKenna, 2004). Generally, higher tides impede discharge, and low tides allow for greater discharge rates, as would be expected. This pulsing of flow creates a mixing zone between the fresh aquifer and the saline marine waters in the sediments (Paulsen et al., 2001). This mixing creates a greater discharge flow quantity than if the fresh groundwater discharged unimpeded into the overlying marine waters, due to the input of salt waters (Martin et al., 2004), and often leads to unfounded concerns because of the disparity between models of fresh groundwater discharges and measurements of discharges to the marine environment. The difference is due to the entrained salt waters (Cable et al., 2004).

However, these processes are not consistent everywhere. For example, no correlation was found between tidal cycling and discharge rates in Great South Bay, potentially because of the exceedingly small (less than one foot) tidal range. Although salt water penetrated the sediments, this was thought to be due to wave-induced effects and salt fingering (Bokuniewicz et al., 2004). In the Florida Keys, submarine groundwater discharges were found to vary seasonally, as in summer tides and groundwater heads are greater. Groundwater heads increase due to seasonal rains, and tides are greater due to lower atmospheric pressures, among other factors (Lapointe et al., 1990). Work in Japan suggested that concave embayments will have discharge rates due to focusing of the groundwater paths, and also found that there was less flow at or near the mouths of rivers. Changes in flow rates close to shore correlated with aquifer head variations, while changes offshore correlated with sea level fluctuations. Aquifer discharges constituted only 15 percent of the total submarine discharge (Taniguchi et al., 2005).

Paulsen has reported on larger-scale measurement techniques that may allow for integrated discharge calculations. Combining resistivity measurements with seepage meter data can lead to inferences regarding bay-wide discharges, as the resistivity readings may be interpreted to delineate some of the important variables discussed just above, such as porosity or the depth of fresh-salt water mixing (Paulsen, 2000).

Theoretical and practical concerns have been raised regarding the data generated by seepage experiments. Generally, the data are thought to more likely to represent true conditions when the device is physically smaller, uses less intrusive techniques to minimize disturbances of the measured environment, is placed where other forces that might affect the results are less likely to affect the measurements, and the overall seepage rate is higher, and more constant (Murdoch and

Kelly, 2003; Shinn et al., 2002). Few marine data sets would meet these criteria, although Murdoch and Kelly thought well of Paulsen's work.

It has been noted that the mixing of salt and fresh waters in these discharge zones often results in altered characteristics of the discharging groundwater compared to the nature of the groundwater measured just onshore (Tsukamoto et al., 2003). It was found, for example, that groundwater discharging into the Peconic Estuary could be responsible for nearly 40 percent of the copper in the bay, although direct measurements of copper concentrations in the groundwater would not have suggested this (Montelucon and Sanudo-Wilhemly, 2001). Furthermore, it is possible that denitrification can influence the amount of nitrate that actually reaches saline waters from the discharging fresh water aquifer (Bratton et al., 2004), and that denitrification is more efficient and effective in the sediments at lower discharge rates (Capone and Slater, 1990). In addition, laboratory work suggests that contaminated marine sediments may transfer organic contaminants into linked fresh water aquifers through a salt pump process, because the salts and organic content of the sea water will drive contaminants preferentially into the fresh water environment (Dror et al., 2003). It is not clear if such contaminants could further migrate inshore in the fresh water system, and improbable they could travel far against typical Long Island shoreline gradients. However, estimates of the impact of groundwater discharge to estuarine water quality often do not account for any such transformations, often because simpler calculations translating groundwater concentrations and flows to estuarine loadings represent considerable advancements in system understandings (e.g., Monti and Scorca, 2003), or because some measurements suggest that good correlations can be made between contaminant concentrations in groundwater and concentrations measured in the estuary, if dilution due to subsurface mixing is properly accounted for (e.g., Gallagher et al., 1996).

1.3 Relationship of Salt Marshes to the Shoreline

Salt marshes are found on the periphery of the shoreline (Chapman, 1960). It is not clear whether they should be classified as "on-shore" or "off-shore" in terms of aquifer discharge. The surface of the marsh tends to lie above mean sea level (Teal and Teal, 1969). The marsh surface can be incised by natural marsh creeks or man-made mosquito ditches. The bottom of natural creeks may or may not lie above mean low water, and so some creeks retain salt water in them throughout the tidal cycle, and some drain completely (Pomeroy and Imberger, 1981). Most

mosquito ditches were designed to drain through tidal cycles, meaning the elevation of their bottoms is above mean low water (Richard, 1938), but this is not necessarily the case on the south shore of Long Island, where micro-tidal ranges mean the typical three foot depth of mosquito ditches can leave their bottoms well below mean low water.

The marsh sediments are often saturated with saline groundwater, as a result of flooding tides. The salty groundwater lies above the fresh groundwater aquifer (Pomeroy and Imberger, 1981). The elevation of the fresh water aquifer has not been published for any salt marsh on Long Island known to us, but presumably is near to mean sea level.

One study on Cape Cod, in a coastal embayment where all of the shoreline was peat except where erosion had occurred, found that all groundwater discharge occurred in a narrow, two meter wide band fringing the seaward side of the marsh (Nowicki et al., 1999). Most other marsh-groundwater theories (see below) tend to describe a greater connection between groundwater discharges and salt marshes, suggesting they do comprise a part of the “offshore zone” insofar as groundwater discharge is concerned.

2. Theories Regarding Groundwater Discharge to Salt Marshes

2.1 General Theories

There are several theories regarding fresh groundwater discharge in salt marshes. One, presented by Howarth and Teal (1980), showed that fresh groundwater discharges occur at the base of the marshes in the bottoms of marsh creeks. The salinity of the salt groundwater system is controlled in these marshes by incidents of tidal flooding and dilution by rain (Teal, 1986). Evaporation may affect summer salinities, leading to elevated salinities in high marsh areas that are not flooded each tidal cycle. Teal's work has been primarily conducted in what are defined as New England salt marshes – the kind of salt marshes found from Maine through Long Island, whose histories were affected by glaciation and subsequent sea level rise.

Pennings and Bertness (1999) described salinity in the marsh soils and aquifer in New England-type marshes as decreasing with distance from the seaward edge of the marsh. This relates to distance from the salt water source, so that fewer inundations by tidal waters means that rainwater constitutes proportionally more of the perched salt marsh aquifer. Alternately, the source of fresh water in the upland edges of the marsh could be groundwater discharges.

Harvey and Odum (1990), working in a fringing marsh in Maryland with a “hillslope” aquifer (the hills were six to 20 m. tall), found that maximal discharge into the wetlands peats was at the upland fringe, and decreased with distance towards the open estuary. Overall, the pore water flows in the marsh were dominated by tidal flows, meaning that groundwater had long residence time in the marsh peats and thoroughly mixed with saline waters prior to discharge through the marsh. The marsh peats, because the base of them is located lower (in relation to mean sea level) than the head of the fresh water aquifer, especially close to the toe of the hill slope, receive discharges from the aquifer. However, residence time in the marsh peats is at least twice as long as in the aquifer sediments. This means, given that there is tremendous input from tidal flows, that fresh water groundwater flow rates in the marsh peats is much slower than in the aquifer sediments, meaning that most flow near the marsh would appear to be under the marsh out into the estuary. This research also found a greater depth of mixing between fresh and salt waters, comparing changing salinities (and specific solutes) downwards through the upper surfaces of the marsh and estuarine sediments. The pattern of change was more even in the marsh as well.

The difference was attributed to the large loss rates caused by root zone processes, and the subsequent replacement of the lost pore waters by tidal flooding or rain. This was thought to be a more vigorous process than the tidal mixing in the off-shore sediments.

A fringing salt marsh was studied in Virginia, where comparisons were made of three different methods of estimating groundwater discharge (Tobias et al., 2001). These were the theory-driven method of considering head differentials and measurements of sediment hydraulic conductivity, a measurement-focused method of comparing salt fluxes flushed from the sediments by the fresher groundwater, and use of a bromine tracer. It was found that the head differential method seemed to be more accurate at lower flow periods, and the salt balance was more accurate at higher flows. Flow rates varied because of seasonal groundwater head differentials, and the greater spring-time groundwater flows, for this system, flushed accumulated materials from the marsh sediments, such as sulfides, and nitrogen and carbon compounds. This was thought to be ecologically important, as in spring many other nutrient inputs may be slowed by lower estuarine system temperatures.

In a low marsh in Massachusetts, discharge from the marsh water table through the banks of creeks was described. The water table losses matched discharge rates in winter, but were twice as great in summer. The difference was assumed to be evapo-transpiration. The seepage was greatest as the tide fell, paralleling the loss of head in the water table. Seepage was much greater for taller creek banks, with 0.5 to 1 meter tall banks releasing three times the volume of porewater as did 0.25 to 0.5 meter tall creek banks (Howes and Goehring, 1994).

In southern marshes, where evapo-transpiration rates are much greater, the salt water aquifers away from the estuary can have elevated salinities above those near the estuary. The salinity of the creek waters is usually the same as the water found in the bankside levees, but the marsh water table water is usually higher in salinity, according to work done in Georgia by Pomeroy and Imberger (1981). They suggested this showed natural creeks drain little water from the marsh, resulting in a consistent-head, perched water table. Similarly, Nuttle and Hemond (1988) found that there was little horizontal flow out of the marshes at creekbanks. Gardner (1975), on the other hand, found that seepage from interstitial sediments enriched the nominally tidally deposited water found on the surface of the marsh with various nutrients, suggesting there was

some sort of hydrological relationship between the saline aquifer and the water that drains off the marsh.

Hemond and Fifield (1982) also thought that seepage in the marsh peat is negligible except near creeks, and theorized that evapo-transpiration is the primary means for removing water from marsh peat away from creeks. Then, due to the loss of head, groundwater inflows would ensue to maintain the perched water table.

Nuttle and Harvey (1995) expanded this argument by constructing a water balance for a marsh controlled by these kinds of flows. Using an assumption no loss of water to the creek from the interior of the marsh, they determined that groundwater upflow volumes were twice as great as tidal inflow volumes, for an irregularly flooded high marsh, because of large evapo-transpiration losses. This theory assumes that at times of large evapo-transpiration rates (when such rates exceed groundwater discharge rates), the salty aquifer will become saltier. When groundwater discharge exceeds evapotranspiration, as during winter, the salty groundwater should become fresher.

They also found that macropores (created by organisms) appear more important for downward transport of tidal waters, and soil matrix pores provide the transport pathway for upward movements of water and solutes. This is because a greater volume of water is stored in the smaller matrix pores, and so evapotranspiration occurs preferentially there (Harvey and Nuttle, 1995).

However, it is not clear that all peats do not transmit groundwater to creeks. A model by Harvey et al. (1987) found that, if the head in the marsh peat layers was great enough, horizontal flows to the creek bank occurred as the tide retreated off the marsh surface. The water balance indicated that two-thirds of the water infiltrating the marsh surface during any particular tide will drain out of the marsh during that same tidal cycle (note the study was made in a shallow, 20 m. wide, *S. alterniflora* marsh that was completely flooded each tidal cycle). Frey and Basan (1985) noted that the greater the height of the tide on the marsh surface, the more infiltration into the sediments would occur, due to greater head. And, generally, infiltration during a high tide is matched by discharge from the sides of tidal creeks during the following low tide; furthermore, it was found that the amount of water infiltrating into the marsh surface decreased with distance

from the marsh creek (Burke et al., 1980), probably relating to reductions in inundation depths. These studies focused on regularly flooded low marshes. Williams et al. (1994), while focusing on high marshes, also suggested that water tables were more variable than consistent. The amount of variation in the water table height would depend on the frequency and duration of flooding, marsh elevation, proximity to and the number of creeks, depressions, and pannes, and the underlying sediment type.

Work in the Everglades, which admittedly is probably not representative of conditions found on Long Island, found a very wide (six to 28 kilometers [km]) mixing zone in the underlying aquifer. Head data collected there suggested that the surface waters in the Everglades (akin to the salt water aquifer in northern marsh peats) were hydraulically connected to the groundwater system. This implies that the mixed groundwater could be a source of brackish water to the surface waters. (Price et al., 2003).

Another interesting, but perhaps only tangentially related process involves marshland upwelling systems (MUS). A MUS is a septic system designed for use in a shoreline setting associated with a salt marsh, where normal septic systems cannot be installed. Injection wells are used to put wastewater into the saline marsh aquifer (various injection sites are used to avoid overloading and pressure channelization). The difference in density between the wastewater and the salt groundwater means the wastewater infiltrates upwards through the peat. This results in treatment of the wastewater, and eventual dilution by tidal flows at the marsh surface (Richardson et al., 2004). This suggests that there must be a relative equalization of pressures between the salt groundwater and the underlying fresh water aquifer that prevents general infiltration and eventual displacement of the salt water groundwater by the fresh water.

Mixing between groundwater and estuarine waters was described as potentially causing chemical changes to the discharging groundwater. Related to those kinds of chemical changes are the findings of Harvey and Odum (1990). Measurements of solutes found that the depth where mixing occurred was deeper for marshes than for estuarine sediments, and that the mixing was more even, in that there was less variation in the salinity and solute measurements. It was suggested that this might have geochemical implications when coupled with the longer residence time for groundwaters in marshes as compared to estuarine sediments, such as allowing denitrification reactions to proceed more to completion. In addition, as might be surmised from

the diversity of microorganisms and redox conditions found in wetlands, it has been shown that biodegradation processes that affect organic contaminants can be greater where groundwater discharges through fresh water marshes than in groundwater alone (Lorah and Olsen ,2000).

It is possible to determine long-term salinity conditions at a marsh. One way is through analysis of the macrophytes preserved in the peat by taking sediment cores. The changes in vegetation can at least be attributed partially to variations in salinity regimes, as different plants can tolerate different amounts of salt (for example, see Orson et al., 1987). As part of this project, the Goodbred laboratory (Marine Sciences Research Center, Stony Brook University) has developed a novel means of quickly assessing marsh histories by comparing photographs of cores made with a Dutch corer to a few select detailed core analyses. This technique allows for inference regarding marsh vegetation to be made on the basis of transects comprised of tens of stations, instead of relying on only a few select core locations, and so makes any determinations more generalized for the marsh as a whole (McLetchie and Goodbred, 2005). Another means of determining marsh salinity history is to analyze diatom remains preserved in the peat, as diatom speciation will vary with salinity, because certain species thrive at particular salinities. Sensitivity of this method appears to be one or two parts per thousand of salinity (Parsons et al., 1999).

Generally, in different settings, different forces may be at work, meaning that it is probably not possible to define one general theory regarding groundwater discharge in or near marshes.

2.2 Implications from Other Work

Bertness et al. (2002) found that increases in nitrogen concentrations, measured in plants, correlate with destabilized marsh vegetative regimes, especially resulting in *Phragmites australis* (*Phragmites*) expansions. They also led to changes in the elevation of the border between *Spartina alterniflora* and *S. patens*, with *S. alterniflora* expanding to higher elevations under higher nutrient conditions. These “excessive” nitrogen concentrations further correlated with the degree of development measured on the upland border of the studied marshes.

The implication of the relationship between developed borders and increased nitrogen supplies to marsh plants is that the development is delivering the nitrogen to the marsh. Generally, on a coastal plain as was the case here, the nitrogen impacts from local development are found in

groundwater. Although not explicitly stated by Bertness et al., it seems to be understood that the local groundwater flow is the source of nitrogen additions to the marsh, and so there must be a hydraulic connection between the shallow flow fresh water groundwater system and the perched salt water system. As discussed above, this is consonant with how some have described the potential for discharge from the groundwater to the salt marsh. In Rhode Island, where Bertness et al. worked, most aquifer systems are shallow, being perched just above bedrock. This may lead to different flow dynamics than on Long Island, where hundreds to thousands of feet of unconsolidated sediments overlie the bedrock.

Valiela et al. (1978) determined that groundwater was an important source of nitrogen to the total nitrogen budget for a Cape Cod marsh. The marsh had springs at its upland reaches; however, the finding depended on groundwater inputs to the marsh through the marsh creek bottoms. This was estimated by comparing incoming tidal salinities with the least salinities measured at ebb flow from the marsh, and determining how much inflow would have been required to dilute the inflow to this level. This seems to greatly overestimate the groundwater contribution, as it is not clear groundwater diluted the entire inflow. Nor is it clear that groundwater inflow occurs as rapidly at high tides (when the marsh surface is flooded, allowing for access to any dissolved nutrients) as the estuarine head is greater, and so is likely to restrict inflows from groundwater. Tobias et al. (2001) clearly thought that this estimation method was inappropriate and impossible to apply at fringing marsh systems, for example.

The conclusions of Valiela et al. have great significance for Long Island estuaries. Because salt marsh systems have been shown in some research to be nitrogen-limited (Valiela et al., 1975), nitrate and dissolved organic nitrogen (the primary forms of groundwater nitrogen) would be transformed to particulate nitrogen by the salt marsh plants. Particulate nitrogen is essential for consumers (Valiela et al., 1978), but it has been hypothesized that changes in the ratio of inorganic to organic nitrogen played a role in the onset of noxious algal blooms in Long Island bay systems (Nuzzi and Waters, 2004).

3 Wertheim National Wildlife Refuge Data

This portion of the Long-Term Plan was commissioned with the thought that the need for use of the Suffolk County Department of Health Services (SCDHS) seepage meter (Paulsen et al., 2000) could be determined. If most discharge of groundwater to salt marshes occurred in relatively well-defined zones, such as the base of creeks and mosquito ditches, then the meter could have great utility in determining patterns of discharge, and perhaps in quantifying the overall relationship between fresh water discharge and saline inputs from the estuary – as has been assayed for individual embayments (Paulsen, 2000). However, the lack of consensus regarding groundwater discharge theories for marshes, the evidence that the phenomenon almost certainly occurs subsurface in interactions between the perched salt water lens and the underlying fresh water system, made use of the meter unlikely to resolve key issues regarding the geographical pattern and temporal variability of discharge.

Particular Long Island marshes do and do not fit the some of the particulars discussed above. *Phragmites* invasions on Long Island began on the East End at the turn of the last century (Lamont, 1997), and those marshes were not especially impacted by development. That is, the early *Phragmites* sites tended not to be developed on the scale seen today, although promising research is underway relating population near to marshes with measures of environmental degradation (S. Goodbred, Stony Brook University, personal communication, 2005). Marshes in parts of the Peconic Bay system and along the North Shore do have hilly uplands, and are therefore likely to have steeper groundwater tables in their immediate vicinity. The steeper slope to the water table suggests a greater chance that the underlying marsh peat will intercept the water table.

The South Shore of Long Island, which generally has a microtidal regime, tends to host marshes with wider expanses of high marsh. In such a setting, it may be possible to determine if variations in water table salinity are due to proximity to uplands, inundations, evapo-transpiration, or rainfall.

Therefore, some local data were collected to determine if it strongly implicated one theory compared to the others, for South Shore marshes. South Shore mainland marshes generally have more mosquito breeding than other areas, and certainly receive more pesticides than marshes

elsewhere on Long Island. It is generally believed that water management can be effective in controlling mosquito populations (CDC, 2001). Understanding fresh water inputs to South Shore marshes would be useful if large-scale manipulations of those marshes are to be undertaken.

Cashin Associates (together with Ducks Unlimited, as a subcontractor), as part of a larger monitoring effort at Wertheim National Wildlife Refuge, has analyzed pore water salinity data from 88 marsh sampling points over a 13 month period from September 2003 through September 2004. The measurements were taken across four distinct regions, totaling 150 acres of marsh, on the east bank of the Carmans River (Figure 1). A protocol established by US Fish and Wildlife Service and USGS as part of a north-east US evaluation of Open Marsh Water Management (James-Pirri et al., 2001) was followed. A soil probe was used to extract water from 15 cm below the marsh surface. The soil probe is constructed of a stainless steel tubing (0.065 cm in inner diameter), 70 cm in length, with one end crimped and slotted to allow the entry of water. A short length of plastic tubing was attached to the opposite end of the probe. Water was drawn up through the probe by a syringe attached to the plastic tubing. Salinity readings were recorded by passing the extracted water through a piece of filter paper placed over the syringe nozzle onto the glass plate of a refractometer. Refractometer readings can be imprecise, especially those made by different individuals. This source of variability was addressed by the measurements being made by the same individual. However, the size of the marsh meant that the readings cannot be considered to be synoptic. In fact, up to four sampling days over a week were required to take all the measurements. In addition, for a variety of reasons, not every station was sampled each sampling event. In all, 14 sampling runs were made (Table 2).

Some general factors regarding conditions during these sampling events were thought to potentially affect the data. One was the occurrence of flooding tides; the other was major precipitation events. Actual tide heights across the marsh were not collected during the time period in question. However, predicted tides from National Oceanic and Atmospheric Administration (NOAA), downloaded using a global positioning system (GPS) unit set for Wertheim co-ordinates, were used as a surrogate. Predicted tide heights ranged from 0.8 feet to 1.1 feet. Tides of one or 1.1 feet were classified as higher tides, and the others were classified as lower tides. It must be understood that for these microtidal environments, storms often influence tidal fluctuations more than astronomical forces. Winds from the north or northeast can prevent

tidal flooding from occurring when it should. Conversely, winds from the south and southwest can pile water deep onto the marsh.

Table 2. Soil Sampling Events at Wertheim National Wildlife Refuge (NWR)

Sampling Event	Year	Day 1	Day 2	Day3	Day 4	Stations Sampled
1	2003	9/30	10/1	10/3		87
2	2003	10/14	10/16	10/17		88
3	2003	10/28	10/30	10/31	11/3	87
4	2003	11/11	11/13	11/14	11/17	86
5	2003	11/24	11/25	12/1		87
6	2003	12/9				43
7	2004	6/7	6/8	6/10		86
8	2004	6/21	6/22			76
9	2004	7/6	7/7			68
10	2004	7/19	7/20	7/21		74
11	2004	8/2	8/3	8/4		64
12	2004	8/16	8/17	8/18		84
13	2004	8/30	8/31	9/1		88
14	2004	9/9	9/10	9/13		88

Precipitation (in the times of the year considered here, rainfall) is also potentially important. However, it is unclear how long a rain event can continue to affect soil salinity. Therefore, rainfall for the sampling events was classified in several ways. Rainfall totals on the sampling date and the day before were used as an immediate measure. In addition, rainfall amounts for the day and the preceding four days, week, and two weeks were collected, using National Weather Service data collected at Upton (approximately five miles north of the site). The data were averaged for each sampling event (as all but one occurred on more than one date). They were classified as in Table 3, based on a rainfall of 0.5 inches being a hard rain for Long Island, and four inches of rain being the average monthly total – with Long Island being considered, generally, a “wet” environment.

Table 3. Rainfall classes, by inches of precipitation

Day and preceding ...	Day	4 Days	Week	2 Weeks
Dry	<0.05	<0.1	<0.25	<0.5
Moderate	0.05<x<0.2	0.1<x<0.3	0.25<x<0.75	0.5<x<1.25
Wet	0.2<x<0.5	0.3<x<0.75	0.75<x<1.0	1.25<x<2.0
Very Wet	>0.5	>0.75	>1.0	>2.0

Other factors received broader classifications. Evapo-transpiration was considered seasonally, with summer having highest rates, and late fall having lower rates. Because the Carmans River constitutes a source of fresh water to the system, it was expected that there might be a salinity gradient from Area 1 south to Area 4, and so the data were examined to see if this generally occurred. In addition, Area 1 lies closest to the forested upland part of the Refuge, and so might be expected to intercept more of the water table than the other areas. Finally, vegetation patterns for each station have been mapped. It is generally thought that *Phragmites* is less tolerant of salt (Bart and Hartman, 2003), and low marsh plants are more tolerant (Bertness, 1991); therefore, salinity patterns in relation to vegetation types were examined. It was anticipated that a good correlation would be found.

Means and standard deviations were calculated for each sampling event (Table 4), and for each of the four areas (Table 5). Table 4 shows there is no temporal pattern to the salinity data, especially in light of the rather large standard deviations. The maximum mean salinity occurred in November 2003, when evapo-transpiration should be lowest, and not under dry conditions – under all four time period classes, the rainfall was determined to have been “moderate.” Rainfall clearly does not control the measured salinities, as each sampling event contained some very low salinities and some salinities that were rather higher than most, no matter whether the sampling event mean tended to be lower or higher than other sampling events. Similarly, Table 5 shows there is no general north-to-south trend in the mean data. The overall mean difference of one ppt between Area 1 and Area 4, in light of standard deviations in the three to four ppt range, seems to indicate there is no real difference between the areas’ salinities. This suggests that neither the estuary nor the river have an overwhelming control on overall soil salinities.

Table 4. Mean salinities for each sampling event (ppt)

Sampling Event	Mean	Standard Deviation	Maximum	Minimum
1	10.9	3.9	23	2
2	13.7	4.5	29	0
3	13.5	4.9	25	0
4	12.2	3.9	21	0
5	15.7	4.7	31	5
6	13.7	5.0	26	0
7	13.6	4.3	22	0
8	12.9	4.8	25	0
9	13.1	3.9	21	0
10	13.5	4.5	25	0
11	13.1	4.0	20	0
12	12.2	4.1	21	0
13	13.4	4.5	25	3
14	12.6	3.7	22	3

Table 5. Mean salinities for each area (ppt)

Area	Number of Stations	Mean of Station Means	Mean of Station Standard Deviations
1	24	12.6	3.3
2	24	13.3	3.3
3	20	13.0	4.3
4	20	13.6	4.3

Statistics for individual stations are collected in Table 6. These data show that some of the station data sets vary from the overall patterns. Mean data, (also presented in Figures 2-5) might be indicated as showing that some of the stations closest to the upland fringe have lower salinities. In Area 1, for example, lower mean salinities were recorded at the first station in transects 1, 2, and 4, and the first two stations in transect 3. However, those mean data are often very similar to the mean salinity for the area as a whole. Only station 1-1-0 was more than a standard deviation different from the area mean, and only 1.3 standard deviations lower. In Area 2, the first stations on transects 1 and 3, and the first two stations on transect 2 are all lower than the Area mean, but none of them were lower by as much as a full standard deviation. The lowest mean salinity in Area 2 was in the middle of transect 1 (2-1-120), and was 1.67 times a standard deviation lower than the mean. The station closest to the Carmans River (2-3-200) had a higher

salinity than other stations – 1.86 times the standard deviation higher. But the station almost as close to the river, station 2-3-120, did not have a salinity much higher than the Area mean. In Area 3, stations closer to the Impoundment and to the north tended to have higher salinities than those along Big Fish Creek and to the south. But the trend was not consistent. This is true although the two stations with means more than a standard deviation different from the Area mean were 3-1-40 (1.15 standard deviations lower than the mean, and located near the Impoundment, to the north) and 3-4-80 (1.83 standard deviations higher than the mean, near the creek, and to the south). In Area 4, the two stations with the lowest salinities (4-1-0 and 4-1-40) were more than one standard deviation less than the mean, and were to the north and east in the Area. The station with the highest mean salinity (4-2-120) was also more than one standard deviation different from the mean, and was near Bellport Bay. But it was not the closest station to the Bay, and while it appeared that the general tendency was for higher salinities near the Bay or Little Fish Creek, these trends were not absolute.

Generally, higher maximum salinities were measured at stations to the south (as measured by Area-wide mean maximum salinities, Table 7). But the difference between Areas was not as great as the inter-Area differences. The mean Area 3 minimum salinities was slightly lower than those for the other Areas. It is interesting that maximum salinities in Area 4 and minimum salinities in Area 3, which represent the two extremes, were more variable than the other data sets.

Table 6. Station salinity data (ppt)

Station	Mean	Std. Dev.	Max.	Min.	Area Mean	Station	Mean	Std. Dev.	Max.	Min.	Area Mean
1-1-0	8.6	3.1	15	4	12.6	2-1-0	10.4	3.7	19	4	13.3
1-1-40	11.4	4.1	20	5	12.6	2-1-40	12.2	4.8	21	5	13.3
1-1-80	11.7	3.6	20	4	12.6	2-1-80	12.4	4.6	25	9	13.3
1-1-120	13.7	2.9	21	8	12.6	2-1-120	10.8	1.5	13	8	13.3
1-2-0	10.4	4.1	16	2	12.6	2-1-160	12.9	2.7	19	8	13.3
1-2-40	13.6	3.5	20	7	12.6	2-2-0	11.6	3.9	19	6	13.3
1-2-80	13.0	5.3	20	3	12.6	2-2-40	11.7	4.8	18	0	13.3
1-2-120	12.9	4.7	18	2	12.6	2-2-80	14.9	3.9	22	8	13.3
1-3-0	10.3	5.5	17	3	12.6	2-2-120	14.6	2.1	18	12	13.3
1-3-40	10.3	4.1	16	4	12.6	2-2-160	12.8	2.9	17	7	13.3
1-3-80	13.9	1.7	16	11	12.6	2-2-200	12.2	5.8	20	0	13.3
1-3-120	13.9	2.2	16	9	12.6	2-3-0	11.5	2.8	16	8	13.3
1-4-160	14.4	1.7	18	12	12.6	2-3-40	14.4	2.7	21	11	13.3
1-3-200	13.3	3.7	20	8	12.6	2-3-80	11.9	2.9	16	8	13.3
1-4-0	12.3	1.7	15	9	12.6	2-3-120	12.4	3.6	17	4	13.3
1-4-40	14.6	2.2	17	10	12.6	2-4-160	11.8	3.2	16	5	13.3
1-4-80	11.6	3.1	16	6	12.6	2-3-200	18.7	2.9	24	13	13.3
1-4-120	12.4	2.4	17	9	12.6	2-4-0	14.1	2.9	18	9	13.3
1-4-160	13.9	3.2	22	10	12.6	2-4-40	13.6	2.1	16	11	13.3
1-4-200	12.9	3.4	21	9	12.6	2-4-80	15.3	3.3	22	11	13.3
1-4-240	13.1	2.6	18	9	12.6	2-4-120	13.9	3.1	19	10	13.3
1-5-0	15.3	2.5	20	12	12.6	2-5-0	18.7	4.4	25	11	13.3
1-5-40	12.4	3.0	18	9	12.6	2-5-40	14.2	3.1	20	10	13.3
1-5-80	11.5	4.3	20	5	12.6	2-5-80	12.8	1.9	16	10	13.3
3-1-0	12.5	3.6	20	9	13.0	4-1-0	9.8	3.6	19	4	13.6
3-1-40	10.7	2.0	15	8	13.0	4-1-40	5.1	4.9	15	0	13.6
3-1-80	10.3	3.3	16	5	13.0	4-1-80	11.6	4.0	21	5	13.6
3-1-120	12.9	2.6	16	10	13.0	4-1-120	14.8	2.8	19	11	13.6
3-1-160	14.4	3.0	20	8	13.0	4-1-160	15.8	3.1	21	10	13.6
3-1-200	13.8	5.8	21	0	13.0	4-2-0	13.9	4.9	26	10	13.6
3-2-0	11.6	4.0	18	5	13.0	4-2-40	14.2	4.3	22	10	13.6
3-2-40	10.3	4.2	17	0	13.0	4-2-80	14.0	3.2	20	8	13.6
3-2-80	11.3	4.1	17	5	13.0	4-2-120	20.2	5.5	29	11	13.6
3-2-120	11.0	2.4	15	8	13.0	4-2-160	18.5	5.8	31	10	13.6
3-2-160	12.1	7.4	21	0	13.0	4-3-0	15.1	3.6	21	10	13.6
3-2-200	14.4	4.2	20	9	13.0	4-3-40	14.3	2.4	20	11	13.6
3-3-0	15.7	4.2	21	10	13.0	4-3-80	14.9	3.3	22	9	13.6
3-3-40	15.3	3.9	20	9	13.0	4-3-120	12.5	6.5	27	6	13.6
3-3-80	14.9	3.0	18	9	13.0	4-3-160	17.6	6.2	25	9	13.6
3-3-120	13.6	8.2	24	0	13.0	4-4-0	11.5	3.4	16	5	13.6
3-4-0	13.1	6.7	21	0	13.0	4-4-40	11.5	7.1	25	3	13.6
3-4-40	11.1	5.8	20	0	13.0	4-4-80	12.3	2.6	15	6	13.6
3-4-80	18.5	3.0	21	10	13.0	4-4-120	11.8	5.7	22	3	13.6
3-4-120	13.8	4.9	21	6	13.0	4-4-160	12.9	4.1	20	8	13.6

Table 7. Mean Maximum and Minimum Salinities, by Area

Area	Mean Maximum Salinity	Maximum Salinity Standard Deviation	Mean Minimum Salinity	Minimum Salinity Standard Deviation
1	18.2	2.1	7.1	3.1
2	19.0	3.1	7.8	3.4
3	19.1	2.4	5.6	4.0
4	21.8	4.3	7.5	3.2

SCDHS took samples of Carmans River water, from the top half-meter of the water column, from four stations (Figure 6) on four dates, three in 2003 and one in 2004. Two samples were taken on three of the sampling dates (morning and afternoon). The data in Table 8 has some similarity to some of the soil salinities. However, the river water appears to be more variable than the soil salinities, although the smaller number of samples make the data inconclusive. Since refractometers have some limited accuracy, the low and high individual salinity readings in the salt marsh soils could reflect salinities associated with the estuarine water immediately offshore from the marsh.

Table 8. Carmans River Salinities

Area	Mean Salinity	Standard Deviation	Maximum Salinity	Minimum Salinity
1	19.2	5.1	27.9	14.8
2	15.0	9.1	27.8	6.0
3	12.1	10.6	27.9	3.3
4	9.2	10.2	27.1	1.1

However, conceptually, the water from the river should only flow out over the surface of the marsh generally during the highest tides (high marsh areas are only inundated irregularly). Higher tides should have higher salinities, as the tidal prism should be composed of saline water from the Bay. This is especially true since salinity data were collected during times of the year when flows tended to be lower, especially for 2004 (Figure 7). Note that none of the major flooding events occurred during sampling (August 17, 2003, and February 7, April 13, and September 29, 2004). In any case, the presence of the weir and its associated impoundment at Southaven Park (north of the Refuge) should tend to equalize fresh water river flows through the estuarine segment of the river.

Many researchers have found correlations between salinity and vegetation cover, as discussed above. The vegetation surrounding each station has been classified into four types:

- High marsh, dominated by *S. patens*, but also including *Scirpus spp.* and other plants
- Low marsh, dominated by tall- and short-form *S. alterniflora*)
- Mixed high and low marsh, where both *Spartina spp.* are present but neither predominates
- *Phragmites*

Again, no clear pattern of soil salinity and vegetation type emerged, even when considered by Area. The mean of the stations means, and maximum and minimum data illustrate that point generally (Table 9). The *Phragmites* data are slightly more variable than the other three vegetation types, but other than that, it is difficult to differentiate between the salinity regimes.

Table 9. Vegetation Type and Salinity Data

Vegetation Type	Number of Stations	Mean of Stations' Mean Salinity	Standard Deviation	Maximum Salinity for All Stations	Minimum Salinity for All Stations
High Marsh	37	12.7	1.9	27	0
Low Marsh	5	14.1	2.3	25	3
Mixed	30	13.4	2.0	25	0
<i>Phragmites</i>	16	13.1	3.3	31	0

It is possible that the mean data are too inclusive a means to reach conclusions from. It may be that the important issue is the pattern of change between results, so that particular stations increase or decrease in salinity similarly according to different conditions. For instance, rainfall may decrease salinity for some stations, but not others, or higher tides may impact some stations only. Multivariate statistics can identify data that vary similarly, or follow particular patterns.

For instance, each sampling result considered can be considered to be a parameter in a multi-dimensional space. Those stations whose data are similar from sampling event to sampling event will plot into the same general area of this multidimensional space, whereas those where the data are most different across the sampling events will plot further away. Such an analysis would use

un-normalized data. A second analysis might look at the way the data varies from event to event, either setting an absolute scale (the lowest and highest salinities expected at the site, for example), or the lowest and highest salinity for the station, or the Area. This kind of analysis would use normalized data. Thirdly, the absolute variation of salinity from event to event could be examined – measuring the gain or loss of salinity for each station. A difference between sampling events would be calculated.

Methods two and three were not considered here, although the reasonableness of each approach may be arguable. Instead, the actual salinities measured at the stations were used as the inputs to the analysis, so that stations where actual salinities were most similar would map closest together. Two multi-variate statistical analyses were conducted, principal component analysis (PCA), which endeavors to construct orthogonal axes to identify factors that generate the multi-dimensional representation of the data, and multi-dimensional scaling (MDS), where distance relations from the multi-dimensional space are reproduced in a reduced dimensional representation. In both cases, two dimensional models were used. Clustered data for both PCA and MDS could indicate which sampling event generated data that were most similar to the data from other sampling events, or which stations varied similarly over time.

Both PCA and MDS require values for all matrix elements. This meant that if the maximum number of stations were used, fewer sampling events could be included. Similarly, including more sampling events minimized the number of stations. Table 9 shows the relationship. This meant that either fewer events would be analyzed to maximize the number of stations, or fewer stations would be included to increase the number of events.

Table 10. Stations and Sampling Events Trade-offs

Number of Sampling Events	Stations Sampled Every Event
14	23
13	38
12	50
11	63
10	69
9	75
7	82
3	88

Unfortunately, none of the analyses that were tried showed any clear clustering tendencies. Neither any of the four rainfall regimes, the two tide classes, or geographical or temporal tracking produced defined clusters of results, whether more events or more stations were considered. Even analyzing the high marsh results in isolation produced no indications that a single factor, such as geography or rainfall was responsible for the data sets. Both PCA and MDS diagrams were very similar, suggesting the results were not biased by selected methodologies. It should be understood that both the PCA and MDS methods accounted poorly for the overall data variability (mostly less than 50 percent), suggesting that the data were difficult to model with only several factors. Cluster analysis, which will group data that are most similar to each other in a tree diagram, so that the main branches identify which groups “belong” together from groups of data that are different, was also tried on several of the data sets. It did not produce any sharper conclusions, as the data tended to be lumped together in an relatively undifferentiated mass.

The Wertheim data show that fresh water inputs occur in the salt marsh aquifer. This is clear from the many zero ppt salinities recorded across the marsh and for almost every sampling event. However, the data analysis could not single out one factor, such as groundwater upwelling or rainfall, as the cause. In fact, the analysis suggests that there is no one factor responsible for pore water salinities for the upper layer of the marsh. Although mean salinity data indicated many similarities exist between stations and over time, which is suggestive that long residence times of tidal inflows mixing with groundwater and rainwater result in a homogenous aquifer, as Harvey and Odum (1990) suggested, there is actually tremendous variation across time and distance in the data set. Thus, the salinity of the marsh water table appears to be a complicated function of all potential impacts – groundwater upwelling, rainfall, evaporation, and inputs from tides – interacting so as to create a heterogenous perched salt water aquifer that is constantly changing.

Thus, it is not clear that changing any one marsh parameter will necessarily impact soil salinities. It also suggests that if manipulation of soil salinities is desired, to reduce *Phragmites* presence for example (although Wertheim data suggest this may be a naïve understanding of conditions that support *Phragmites*), that it may be difficult to achieve desired ends intentionally. This is because the controls on soil salinities in these low tidal amplitude environments appear to be very complex.

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